Loss of remnant trees causes local population collapse of endemic Grosbeak Starling *Scissirostrum dubium* in Central Sulawesi, Indonesia

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Large and isolated trees are often last refuges for rare forest species in highly fragmented and human-dominated landscapes. This is of particular importance in tropical forest margin areas where remnant forest trees are being cleared at an alarming rate. Drivers and consequences of such remnant forest tree losses are still poorly documented. Here we report the rapid destruction of remnant trees, closely associated with colonies of the Sulawesi endemic Grosbeak Starling *Scissirostrum dubium*, which excavates nest holes in large dead trees. In 2008, we mapped all the species's potential breeding trees, tree characteristics and the local population density on the east margin of Lore Lindu National Park, Central Sulawesi, Indonesia. When the area was revisited in 2010, we found a dramatic loss of 92% of the recorded nest sites, accompanied by a remarkable decline of the local Grosbeak Starling population. This study provides an alarming example of the immediate consequences of the loss of remnant forest trees in tropical human-dominated landscapes for species dependent on this habitat structure. Without the contemporary implementation of strategies maintaining a high density of isolated large trees in forest margin zones and adjacent cultivated areas, associated species will experience dramatic population declines and a high local and, in the mid- to long-term, a high regional risk of extinction.

INTRODUCTION

Deforestation degrades habitats and isolates populations, thereby substantially reducing biodiversity (Brook *et al.* 2003, Gardner *et al.* 2009). South-East Asia, a region of high biodiversity holding an exceptionally high number of endemic species, is threatened by the highest rate of human-caused habitat loss (Sodhi *et al.* 2004, Sodhi *et al.* 2010a,b). The ecological impacts of deforestation are poorly understood in many parts of South-East Asia (Sodhi *et al.* 2005), including Sulawesi, the largest island in Wallacea, which is characterised by an extensive endemic avifauna (Stattersfield *et al.* 1998, Lee *et al.* 2007). Many South-East Asian bird species suffer from a lack of protected reserves and conservation funding as well as from intense human encroachment into their habitat (e.g. Whitten *et al.* 1987, Sodhi *et al.* 2010a).

In this study, we focused on large remnant trees at the forest margin and the adjacent cultivated area of Lore Lindu National Park (hereafter NP), which are increasingly threatened by habitat modification and land-use intensification. Large remnant trees are important structural elements for birds in human-modified habitats (Abrahamczyk *et al.* 2008). However, the conversion of semi-natural habitats into intensively used agricultural land results in population declines in many different species groups (Berry *et al.* 2010). Given the rapid and ongoing conversion of natural habitats into human land-use systems in Sulawesi, improvement in forest management and conservation is overdue to prevent the extinction of endangered and endemic species (Sodhi *et al.* 2005, 2010a, Miettinen *et al.* 2011). This includes improved understanding of largely unknown species and their habitat requirements; in the case of the Sulawesi endemic Grosbeak Starling *Scissirostrum dubium* data are very limited (Coates & Bishop 1997, Craig & Feare 2009). This species utilises large and isolated dead trees for nesting and foraging, and is mainly found on the margins of forests and in lightly wooded areas. It is a colonial breeder and the only starling known to excavate its own nest-holes in tall dead or rotten trees. The species forages in small groups, mainly feeding on a wide range of fruiting trees (e.g., Fabaceae, Moraceae, Myrtaceae) as well as insects and seeds (BM unpubl. data). Colonies comprise highly social flocks of 20–150 birds (Coates & Bishop 1997), which may form 'super-colonies' of more than 1,000 individuals in several adjoining trees. It appears that the species was originally a forest-dweller that has adapted to forest margin habitats very well.

In 2008 and 2010, we quantified the large remnant forest trees on the east margins of Lore Lindu NP, Central Sulawesi, associated with a Grosbeak Starling population. The study focused on (1) mapping the remnant forest trees and Grosbeak Starling colonies in the forest margin zone in 2008 to identify important characteristics of nesting trees and habitat requirements, (2) based on the mapping, the species's population density was assessed and (3) in 2010 the mapping of colonised trees was repeated to quantify the extent of remnant forest tree decline (involving the loss of nesting sites) and to re-evaluate the species's current conservation status in the area.

METHODS

Study area

The study site was about 75 km south-east of the provincial capital Palu. The area was declared a UNESCO Man and Biosphere Reserve in 1977 and the 229,000 ha national park was established in 1993 (Adiwibowo 2005). It is an exceptionally species-rich area holding about 78% of Sulawesi's endemic birds (Coates & Bishop 1997). The forest-margin landscape outside the closed block of near-primary forest is a mosaic of secondary forest and a rapidly increasing number of land-use systems—cocoa, coffee, maize and rice being the main crops (Schulze *et al.* 2004, Maas *et al.* 2009). Remnant forest trees were mapped in an area of about 45 km^2 at the northern end of Napu Valley near the villages of Wuasa (1.426°S 120.315°E), Alitupu, Kaduwaa, Banyusari and Watumaeta, in the lower montane forest zone (Whitten *et al.* 1987) between 1,100 and 1,200 m, an area with mean annual rainfall of over 3,000 mm.

Data collection

The first mapping of remnant forest trees, including tree measurement, observation of Grosbeak Starling colony trees and an estimate of the species's local population density, was carried out between 22 August and 22 September 2008. All the colony trees were re-visited between 1 April and 5 May 2010. Mapping was done between 06h00 and 19h00 daily in both periods. We also mapped the forest margin at 158 GPS points in 2008 and re-mapped it in 2010, taking additional data at locations where the forest margin had changed due to conversion, logging and land-use expansion. Throughout the study, the code of ethics of the American Birding Association was observed (www.aba.org/about/ethics.html).

Tree mapping was carried out using a geo-referenced map of the study area (Bakosurtanal 1991), GPS data (Garmin 12 Map) and a digital rangefinder to measure and estimate distances (Nikon Laser 800S). In 2008, all trees with a diameter at breast height (dbh) greater than 20 cm and more than 2 m tall outside the closed forest margin area (using 50×50 m grids within the study area) were mapped. We also mapped all trees formerly or currently occupied by Grosbeak Starling inside the forest, on 22 forest transects, 300 m long and 200 m wide, with buffer zones 100 m left and right of each transect. Each of the 547 trees recorded was allocated an individual number, a GPS position and plotted on the map. For each tree, we recorded the height (estimated using a digital rangefinder), dbh, distance to primary forest (GPS data), date when the area around it was last logged (3 categories: less than 5 years, 5–10 years, more than 10 years), the habitat type (agricultural, primary or secondary forest) and the number of Grosbeak Starling breeding holes; these have a characteristic shape, easily distinguished from other tree-holes (e.g. created by arthropods or other animals). Breeding holes were not necessarily in active use by Grosbeak Starlings, and we also checked whether the trees (n = 24) with breeding holes were visited by the species in 2008—twenty trees were in use. Almost all the trees lacked identification characters (e.g. leaves, branches) or were dead or rotten and could not be confidently identified. In 2010, we re-assessed the 24 colony trees recorded in 2008. All mapping data were analysed using ArcGIS (ESRI 2005).

We determined the number of Grosbeak Starlings in each occupied tree. The number of available breeding holes on each tree was counted, then each occupied tree was watched for six hours (divided into 2–3 observation periods) and the number of active holes determined. These observations, carried out by BM and an experienced local guide, also served to obtain additional information on behaviour, breeding activity and foraging times. Each occupied hole was used by two adult Grosbeak Starlings (and occasionally their chicks). Accordingly, the number of active breeding holes multiplied by two provided a conservative estimate of the population, including only adults which regularly returned to their holes. The long observation period at each tree ensured that holes not occupied by Grosbeak Starlings were correctly identified. The population was recorded on a tree-by-tree basis, and if the birds occupied more than one tree at a location it was designated a 'super-colony'.

Statistical analyses

To determine the importance of individual parameters in colonisation by Grosbeak Starlings, we analysed tree parameters by calculating Pearson correlations to relate tree height (m) and tree dbh (cm) to the number of breeding holes for each colonised tree. The distribution of the response variable was either assumed to be approximated by normal (tree height, tree dbh) or overdispersed Poisson (number of breeding holes) distribution. In the last case, variables were log-transformed. In addition, the tree parameters were divided into five dbh classes with 50 cm intervals and seven tree height classes with 5 m intervals and tested against the respective percentage of trees colonised within the intervals using Spearman rank correlations. Tree occupation by the species was displayed within a bubble plot with colony size as weighted variable (colony size intervals of 20 individuals) against both tested tree parameters (tree height and tree dbh) on the plot axes. All statistical tests and two-dimensional plots were computed using STATISTICA version 7.1 (StatSoft Inc. 2005).

RESULTS

Tree mapping and Grosbeak Starling nesting requirements

We mapped 547 trees in 2008, and identified Grosbeak Starling breeding holes in 24 trees, with 20 trees in active use for nesting (Table 1). Correlation between tree height and dbh was highly significant for all 547 mapped trees ($r = 0.412, p < 0.001$), significant for all 24 trees with breeding holes ($r = 0.456, p = 0.025$) and not significant for 20 occupied trees ($r = 0.269, p = 0.252$). Larger Grosbeak Starling colonies occupied larger trees. The number of breeding holes per tree (log) was correlated significantly with the dbh of colonised trees (Figure 1A) but not with their height (Figure 1B). However, the percentage of colonised trees increased with both increasing size of dbh class (Figure 1C) and tree height class (Figure 1D).

Trees with breeding holes were all more than 16 m high, with a minimum dbh 43.3 cm. The majority of occupied nesting trees (75%) had a dbh greater than 80 cm (Figure 2). Besides the preference for large trees, the species also preferred to make its holes in the upper trunk. The minimum height of an occupied breeding hole was 11 m. Some of the occupied trees made up 'super-colonies' of up to three trees in one location. Trees with breeding holes were located in secondary forest habitat (42%), agroforestry systems (33%), vegetable fields (12.5%) and other intensified land-use systems (> 5%). The sites of 67% of the trees were last logged more than ten years previously while the rest were split equally between sites cleared, leaving only a single tree, in the last ten or five years. Of the 24 trees with Grosbeak Starling breeding holes recorded in 2008, only two remained in the re-mapping survey of 2010. These two trees had formed a 'super-colony' consisting of three trees which remained in uncultivated land of complex topography and difficult access owing to water-logged areas. All other 22 former colony trees were lost as a result of logging and wildlife trading activities—trees were cut down either to collect Grosbeak Starlings or to make land-use changes, or else they had collapsed naturally. However, it was very difficult to determine specifically the fate of individual trees. Information obtained locally indicated that some trees became so rotten that village farmers were able to pull them down without

Table 1. Trees with Grosbeak Starling breeding holes in 2008. Trees 1–20 held active colonies; trees 21–24 had nest holes but had been abandoned. The height of the lowest occupied breeding hole and the total number of breeding holes on each tree are shown. Habitat type: PF primary forest, SF secondary forest, AF agroforestry, OL open land: annual crops and pasture, VF vegetable field. *the trees still standing in 2010.

Tree	Colony size	Height (m)	Dbh (cm)	Habitat type	Distance (m) to primary forest	Last logged (years)	Height of lowest hole (m)	Total breeding holes
1	80	22	166.79	SF	800	> 10	15	123
2	60	32	124.14	SF	800	> 10	22	179
3*	55	27	132.74	SF	800	> 10	19	116
4	50	35	80.53	SF	180	> 10	20	7
5*	50	38	210.40	SF	500	> 10	24	171
6	40	18	89.45	SF	85	> 10	12	19
7	40	30	52.84	AF	60	> 10	NA	36
8	40	24	140.37	OL	500	5–10	14	32
9	40	16	67.48	OL	3000	> 10	14	33
10	30	17	99.63	VF	700	5–10	12	32
11	20	20	53.16	SF	800	> 10	15	25
12	20	40	43.29	SF	500	> 10	32	6
13	18	30	77.35	SF	10	> 10	NA	15
14	18	35	128.92	PF	−300	> 10	NA	35
15	18	27	83.72	AF	250	5–10	24	28
16	18	23	88.81	AF	90	> 10	11	19
17	16	16	72.57	VF	400	0–5	NA	23
18	6	33	162.34	VF	1100	0–5	31	4
19	6	24	81.81	AF	320	0–5	19	5
20	6	22	89.45	SF	2750	> 10	NA	20
21	0	16	51.88	AF	200	0–5	NA	27
22	0	15	27.06	AF	100	5–10	NA	18
23	0	18	56.66	AF	3000	> 10	NA	5
24	0	9	48.38	AF	3200	> 10	NA	7

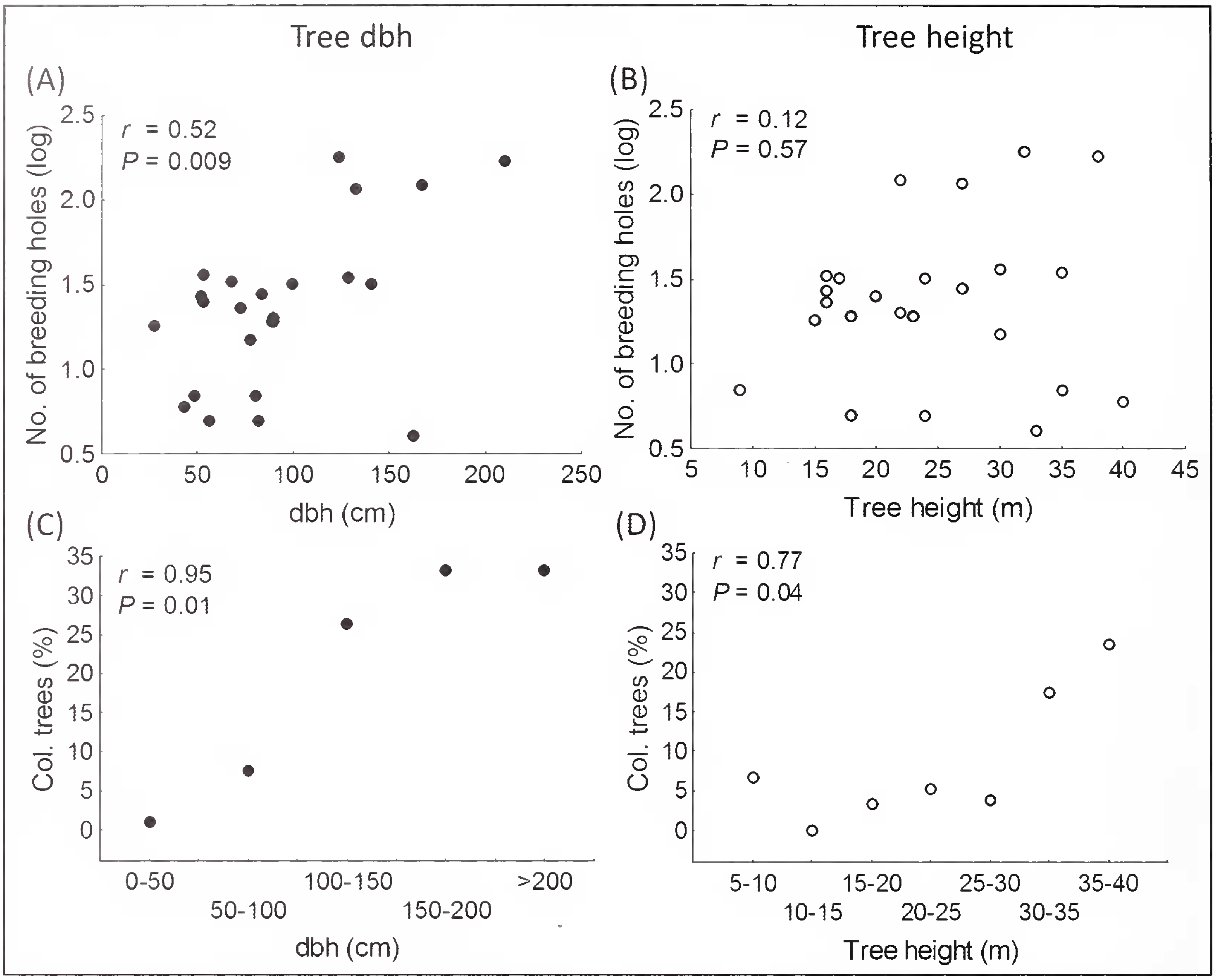

Figure 1. Effects of tree dbh and tree height on colony size and the proportion of colonised trees. (A) The number of breeding holes is correlated significantly with the tree dbh (cm) of colonised trees (n = 24) whereas (B) tree height is not related to colony size (results of Pearson correlations). The percentage of colonised trees of different tree size classes (x axis) significantly increases with increasing tree size although the effect is stronger for dbh class (C) than for tree height class (D) (results of Spearman rank correlations).

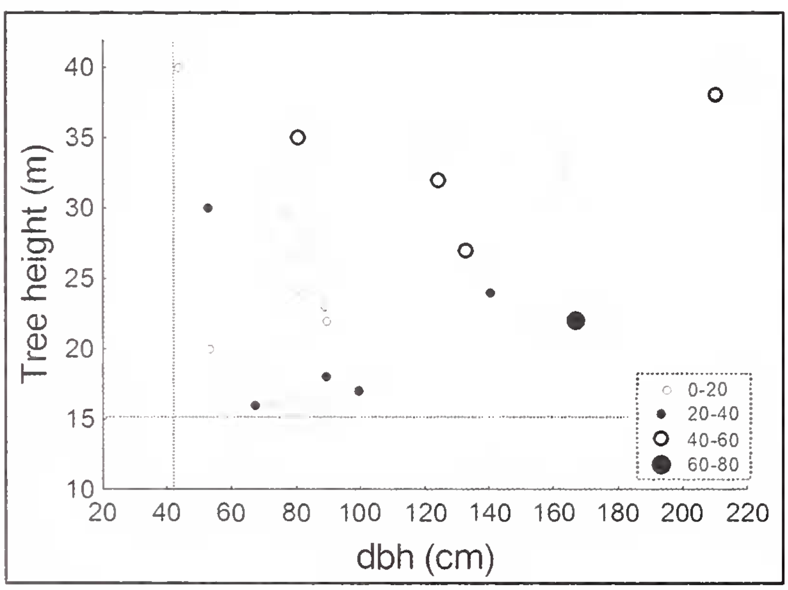

Figure 2. Bubble plot of tree height (m) and dbh (cm) of occupied trees (n = 20) with increasing Grosbeak Starling colony size (intervals of 20 individuals per tree) indicated by circles of increasing size. Only trees outside the dashed lines (taller than 16 m and dbh greater than 43.3 cm) were occupied.

recourse to use of saws or axes before they fell naturally, and in one case the area was burnt. Several areas were converted to cacao plantations and young shade trees were planted in place of the relict forest trees, whilst limitations in accuracy of GPS did not allow the location of erstwhile nesting trees to be pinpointed. It was confirmed that two trees were cut down specifically to obtain the birds which were subsequently sold, but it cannot be ruled out that trees that were lost in cacao plantations or pulled down manually by villagers were also taken down to obtain birds and sell them.

Grosbeak Starling population and biological data

A total of 684 Grosbeak Starlings were observed at the 20 occupied trees within 45 km^2, corresponding to 15 individuals per km^2. The mean colony size per tree (± SD) was 26.29 (± 22.06) birds. We also recorded breeding activity—chick feeding—in August and September 2008. Previously, breeding had apparently only been reported in May (Craig & Feare 2009).

DISCUSSION

Our results document the rapid local decline of large remnant forest trees and the loss of 92% of Grosbeak Starling nesting sites in only two years. The rapid decline of the Grosbeak Starling population in our study area indicates that suitable nesting sites are likely to become the limiting factor for this species's survival, at least at local level. In 2008, four of the 24 trees with breeding holes were not in active use. This may have been due to a sufficiency of nesting sites at that time, or to abandonment due to high parasite densities or anthropogenic disturbance (the four trees were in agroforestry systems frequently used by the local community).

Deforestation and the intensification of land-use systems are ongoing in Central Sulawesi and elsewhere on the island (Sodhi *et al.* 2012), putting pressure on the remaining local Grosbeak Starling populations and other species associated with remnant forest trees (Gardner *et al.* 2009). The long-term effects of remnant forest tree losses and deforestation on biodiversity and ecosystem resilience (e.g. soil quality) are largely unknown, but there is evidence that the resulting species extinctions can continue for many decades (Brooks *et al.* 1999) and that this loss of biodiversity might be irreversible (Dupouey *et al.* 2002).

Grosbeak Starling, a forest species adapted to the forest edge, depends on tall, large standing trees at the forest margin or in the adjacent cultivated area for establishing breeding colonies. We found that trees suitable for Grosbeak Starling could best be described using the dbh of colony trees (minimum 43.3 cm), which was positively related to the number of breeding holes per tree. Furthermore, the proportion of trees colonised increased rapidly with increasing dbh, and most nesting trees (75%) were more than 80 cm dbh. Tree height was a rather poor indicator for colonisation, probably because thick and mainly rotten, often dead, trees were frequently broken at the top.

The majority of trees colonised were in secondary forest and agroforestry systems as well as in areas last logged ten years or more previously. Secondary forest and logged forest can hold high bird species diversity, although species abundance declines as a consequence of logging (Berry *et al.* 2010). The removal of remnant trees and shade trees in human-dominated landscapes is also likely to increase the risk of pest outbreaks and leads to a loss of the multiple ecosystem services provided by these trees, including food and non-food resources, carbon storage, nutrient cycling and erosion control (Bhagwat *et al.* 2008, Tscharntke *et al.* 2011). Beside these very valuable ecosystem services, the trees are often utilised by native forest species in areas adjacent to remaining forest fragments, as in the case of the Grosbeak Starling.

However, tolerance of anthropogenic activities becomes a disadvantage for species negatively affected by land-use intensification (e.g. Sodhi *et al.* 2012, Newbold *et al.* 2013) as well as those species that are hunted, traded and removed from their nesting trees (Fuller 2002, Wilcove *et al.* 2013, BM unpubl. data, K. Darras pers. comm. 2014). The Grosbeak Starling population in Sulawesi, as well as an increasing number of bird species in South-East Asia (many of them forest margin species, habitat specialists and species with poorly documented habitat requirements and responses to habitat fragmentation), are facing multiple threats associated with remnant tree loss (e.g. Maas *et al.* 2009, Bregman *et al.* 2014). Major threats include illegal logging and hunting activities (e.g. Fuller 2002), with potential consequences for ecosystem resilience and ecosystem services in areas with high anthropogenic disturbance (Sethi & Howe 2009). Hunting pressure has become a serious additional threat to Grosbeak Starlings owing to their increasing attractiveness for the pet trade—caged birds for sale were frequently observed in Sulawesi, Java and Jambi, Sumatra, between 2009 and 2014 (BM & CHS pers. obs., K. Darras pers. comm.), and because they are frequently confused with agricultural pest species such as Short-tailed Starling *Aplonis minor* by the local community (BM pers. obs.). Inadequate law enforcement in maintaining protected areas and the rapid ongoing loss of suitable nesting trees seem to be the most immediate threats to this species. The communication and integration of research results into awareness training of local communities represents a promising contribution to the conservation of endangered biodiversity in the tropics (Laurance 2013), especially in smallholder-dominated areas (Persha *et al.* 2011) such as Central Sulawesi. Furthermore, landscape management practices in the tropics should take into account the high potential of forest patches, extensively used agroforestry systems and vegetation corridors to support high levels of species diversity (e.g. Harvey & Villalobos 2007, Bregman *et al.* 2014), and to allow species recolonisation after local extinctions (e.g. Gillies *et al.* 2011).

Large remnant trees represent important structural elements for birds in human-modified habitats in the tropics (Abrahamczyk *et al.* 2008), as well as for many other species (e.g. Gardner *et al.* 2009). Although our recorded trees could not be identified with complete certainty, the majority may be the species hassk *Erythrina subumbrans*, a common species in our study area, native to India and Sri Lanka and today widely distributed, or the somewhat larger hassk *Parkia speciosa*, native to Sulawesi; both are used as shade trees for cacao crops. Available studies from Neotropical forests illustrate the high variation of age (~15 to 115 years) of tree species with dbh values higher than 40 cm (Schöngart 2008, Leoni *et al.* 2011). It is well understood that tree growth rates also strongly vary depending on the species and location-specific characteristics. Trees in secondary forests and agroforestry systems of Central Sulawesi 15 m or more in height are often older than the respective forest succession stage or the land-use systems themselves, and were left behind during ongoing forest conversion process (M. Kessler pers. comm. 2013). Translated into the habitat requirements of the Grosbeak Starling, an even longer and yet unknown period than the minimum available estimate (15 years) would be necessary to plant and grow new trees to provide future nesting sites for their colonies, since the species can only excavate breeding holes in trees which are already dead or rotten and therefore easy to perforate. Consequently it mostly occupies tree remnants formerly part of the forest interior but now at the forest edge owing to previous slash-and-burn activities. The other category of trees colonised by Grosbeak Starlings persists in patches of older, uncultivated secondary forest or agroforestry sites in the open land area, adjacent to the eastern border of Lore Lindu NP. Hence, to depend on ageing secondary forest habitats or extensive agroforests appears to be an unsuitable strategy to avoid further tree declines and local Grosbeak Starling extinctions in the near future.

CONCLUSIONS

The loss of remnant large trees in tropical forest margins is a serious, probably underestimated threat to associated forest species. Our findings on the eastern border of Lore Lindu NP are consistent with an alarming trend in many tropical forest margin landscapes (Bregman *et al.* 2014). Until now, the population size of Grosbeak Starling has not been quantified (BirdLife International 2015) and more evidence is needed to determine whether the clearance of breeding trees and the collapse of nesting sites is widespread. Our study provides an example of the immediate consequences of remnant forest tree losses that potentially affect many different species (e.g. Gardner *et al.* 2009, Bregman *et al.* 2014) and underlines the importance of direct observations of the effects of habitat conversion and land-use intensification on associated biodiversity. The risk of indirect observations is that rapid changes in the associated species community might be missed or largely underestimated.

Increasing the understanding and valuation of ecosystem services and biodiversity functions among rural and local smallholder communities can decelerate the rapid loss of remnant forest species in Central Sulawesi and may provide a promising conservation strategy in many protected areas in the tropics (Persha *et al.* 2011, Laurance 2013). The protection of large trees (such as the widespread *Erythrina* shade trees) in the human-dominated rainforest margins of Lore Lindu NP is necessary to maintain stable population densities of Grosbeak Starlings, at least in the short to mid-term.

Wilcove *et al.* (2013) point out the major challenges of improving biodiversity-friendly concepts in cash-crop producing areas; they show that the financial returns from logging and palm oil production in lowland Sabah, East Malaysia, are, on average, twice as large as those currently possible under conservation measures (incorporating payments for carbon, biodiversity, water and ecotourism). However, this calculation does not account for the negative long-term effects of habitat transformation, forest fragmentation, land-use intensification and biodiversity loss which are likely to affect ecosystem services such as biological control and seed dispersal (e.g. Sethi & Howe 2009, Bregman *et al.* 2014) and therefore to result in tremendous economic impacts in the future (Kellerman *et al.* 2008, Karp *et al.* 2013, Maas *et al.* 2013), especially affecting smallholder plantation owners and being potentially irreversible (Dupouey *et al.* 2002, Lindenmayer *et al.* 2006).

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